To: Laidlaw, Tina[Laidlaw.Tina@epa.gov]
From: Suplee, Mike
Sent: Wed 4/2/2014 8:40:15 PM
Subject: Memos
TECHMEMO_14Q5_FNL.pdf
TECHMEMO_BenthicAlgaeThreshold_FNL.pdf

;
Hi Tina;

Here are the two technical memos.

MIke

Montana Department of
ENVIRONMENTAL QUALITY

MEMO

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To: Montana Board of Environmental Review

CC: Eric Urban, Water Quality Standards Section Supervisor

From: Michael Suplee, Ph.D., and Kyle Flynn, P.H., Environmental Science Specialists, Montana Dept. of Environmental Quality

Date: 3/19/2014

RE: Derivation of the Seasonal 14Q5 Low-flow Design Flow for Wadeable Streams and Large Rivers

When Montana Pollutant Discharge Elimination System (MPDES) permits are developed, a low-flow design flow is routinely used to calculate a permittee's allowable discharge concentrations. The Department uses a seven-day, ten-year design flow (7Q10) for permitting pollutant discharges (ARM 17.30.635(2)). But this low-flow was designed primarily for toxics and has year-round application. Existing rule directs the Department to identify a low-flow design flow specific to nutrients (ARM 17.30.635(2)). Therefore, we explored alternative design flows that might be more appropriate for discharges containing nitrogen and phosphorus (nutrients) and which could be applied seasonally to the proposed base numeric nutrient standards in MAR notice No. 17-356.

The result of our work was the seasonal 14Q5 low-flow design flow. Rules (e.g., ARM 17.30.635) have been modified to include the seasonal 14Q5 and these modifications are also found in MAR notice No. 17-356. The seasonal 14Q5 low-flow is **specific to discharges containing nutrients**. The purpose of this memo is to describe the process by which this nutrient-specific low-flow design flow was developed.

Development of the 14Q5 Low-flow Design Flow for Discharges Containing Nitrogen and Phosphorus

The most important low-flow design period for nutrients is the summer and fall baseflow period (growing season), when water quality is most likely to be impaired by excess nutrients. Streams in Montana tend to reach stable baseflow, elevated temperatures, greatest water clarity, and maximum photoperiod at about the same time, beginning in late June or early July (Suplee et al., 2007). In large rivers this period generally begins later, usually around August 1st (Flynn and Suplee, 2013). The point in time in the fall/early winter when this growing season ends is somewhat subjective, but based on rapidly declining temperatures, diminished light levels, etc., sometime in October is probably appropriate. Given these considerations, then, the growing season is the most logical time for the application of nutrient standards and a seasonal low-flow design flow.

Algal growth rates govern the time required to reach a given algal biomass. They are dependent on temperature, light, and nutrient limitation, and are the precursor to all the attendant eutrophication responses. It is therefore necessary to constrain loadings (i.e., limit nutrient concentrations) over durations when nuisance growth and associated water quality excursions are expected to (and can physically) occur. Since bottom-attached (benthic) algae have been shown to be very influential to river and stream primary productivity (Stevenson et al., 1996), benthic algal growth rates from the literature (**Table 1**) were used in conjunction with a simple analytical model to derive a suitable duration for appraisal of river and stream water-quality. The model (presented below) provides a low-flow design flow supportive of river beneficial uses.

Table 1. Enrichment Studies and Associated Net-specific Growth Rates Adjusted to 20 Degrees C. Growth rates were corrected to the reference temperature using the Arrhenius equation (Chapra et al., 2008).

Algae Type	Net Specific Growth Rate at 20°C (k, day^{-1})	Reference	Location	Comment
Diatoms	0.50	Klarich (1977)	Yellowstone River, MT	Near Huntley Billings WWTP
Diatoms	0.55	Bothwell and Stockner (1980)	McKenzie River, OR	5% kraft mill effluent
Cladophora	0.71	Auer and Canale (1982)	Lake Huron, MI	Harbor Beach WWTP
Green algae	0.52	Horner et al. (1983)	Lab Flume	Laboratory N & P addition
Diatoms	0.42	Bothwell (1985)	Thompson River, BC	Downstream of WWTP
Diatoms	0.62	Bothwell (1988)	S. Thompson River, BC	Flume with N & P addition
Diatoms	0.58	Biggs (1990)	South Brook, New Zealand	Downstream of WWTP
Diatoms	0.45	Stevenson (1990)	Wilson Creek, KY	Agricultural stream after spate

Growth of stream and river benthic algae typically follows a general pattern of colonization, exponential growth, and autogenic sloughing and loss (Stevenson et al., 2006). The net accrual portion (i.e., colonization and growth) can be readily modeled using a first-order exponential net growth equation (**Equation 1**), with space limitation (**Equation 2**), per Chapra et al. (2010),

$$\frac{da_b}{dt} = a_b \phi_{Sb} k \tag{1}$$

$$\phi_{Sb} = 1 - \frac{a_b}{a_{b,\max}} \tag{2}$$

where a_b = benthic algal biomass (mg Chl*a*/m^2), ϕ_{Sb} = a space limitation factor (dimensionless), k = temperature dependent first-order net-specific growth rate (day^{-1}), and $a_{b,\max}$ = maximum biomass carrying capacity (mg Chl*a*/m^2). Equations 1 and 2 can be combined and solved analytically (**Equation 3**),

$$a_b(t) = \frac{a_{b,\max} \exp^{kt}}{\frac{a_{b,\max}}{a_{b,init}} + \exp^{kt} - 1} \tag{3}$$

where $a_b(t)$ = benthic algal biomass (mg Chl*a*/m^2) at a defined point in time after growth initiation, $a_{b,init}$ = initial biomass condition (mg Chl*a*/m^2), and t = time (days), so that peak biomass (PB) and time to peak biomass (T_{PB}) (per Stevenson et al., 2006) can be identified (**Figure 1**). For the design flow determination, an initial biomass of 0.1 mg Chl*a*/m^2 was assumed for all growth calculations with an $a_{b,max}$ of 1000 mg Chl*a*/m^2 (Horner et al., 1983).

We further define additional points of interest in the accrual curve (**Figure 1**), namely nuisance biomass (NB) and time to nuisance biomass (T_{NB}). These occur between the initial colonization phase and PB and reflect the point where a nuisance response would occur absent of nutrient limitation. For a nutrient control s ... ed so that PB never re

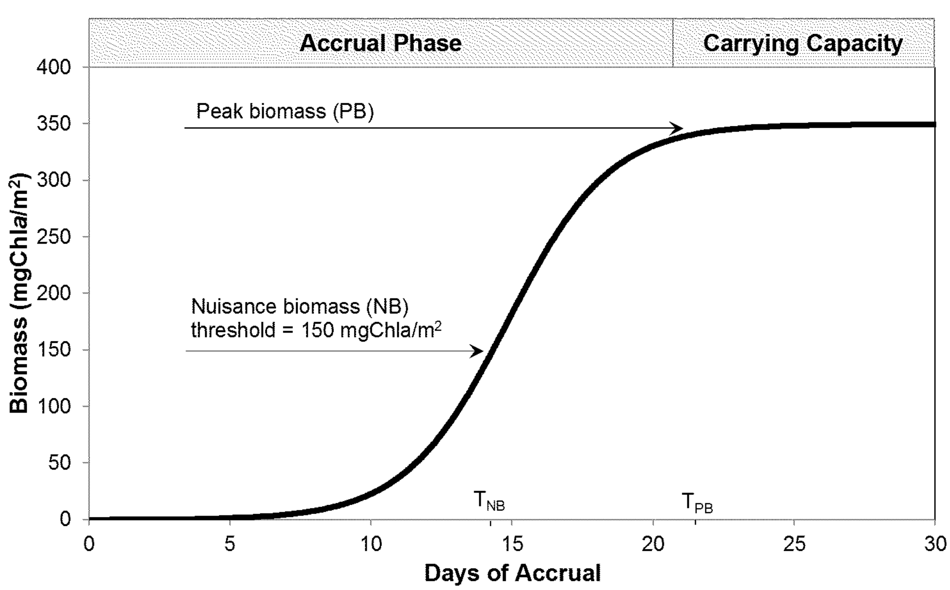

Figure 1. Modeled Accrual Phase for Benthic Algae Showing Colonization, Exponential Growth, and Peak Biomass. Time to peak biomass and nuisance biomass are shown on the abscissa.

Studies with moderate nutrient enrichment and time-variable benthic algal biomass measurements were compiled (**Table 1**) so that T_{NB}, PB, and T_{PB} could be estimated for nutrient concentrations similar to the proposed standards. We only considered studies that reported water temperature so that corrections to a standard reference temperature (20°C) could be made, and the Arrhenius equation was used to make these adjustments (Chapra, 2008). Light was not believed to be a limiting factor in the compiled studies. Temperature-normalized growth coefficients (*k*; day^{-1}, 20°C) averaged 0.55 ± 0.09/day (95% confidence level) and, using **Equation 3** above, yielded times-to-nuisance biomass from 11-17 days, with an average of 14 days (**Figure 2**). Fourteen days closely matched the T_{NB} determined for the Yellowstone River (Klarich, 1977) and was selected as the duration interval. The 14-day T_{NB} applies to shallow areas (< 0.5 m) of large rivers, as well as to wadeable streams which normally have shallow depths in summer.

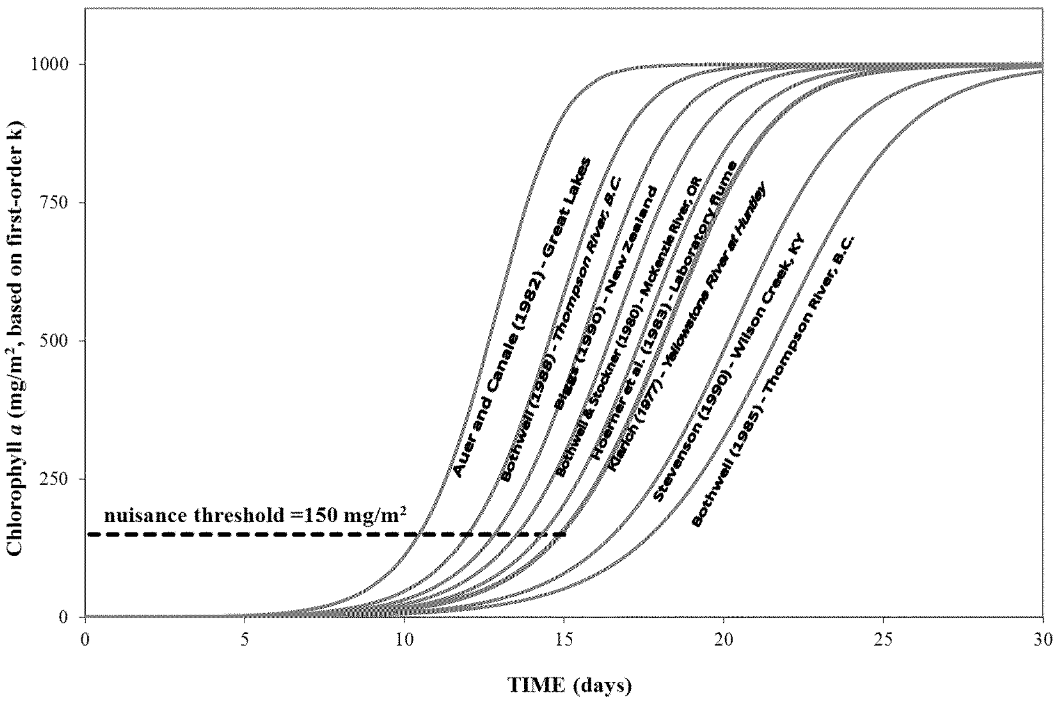

Figure 2. Estimated Time to Nuisance Algal Biomass under Moderately Enriched Conditions. Each curve was generated using the k values in Table 1. Time to nuisance biomass was approximately 14 days.

The time to nuisance biomass estimate (14 days) could actually be lower or higher than 14 days and warrants further consideration. The time to nuisance depends, in part, on the initial biomass used. It is possible that the initial biomass we used for the growth curves (0.1 mg Chl*a*/m^2; **Figure 2**) was too low, and the algae standing crop more common in summer (5-50 mg Chl*a*/m^2) would rise to a nuisance level more quickly than estimated. But if the proposed nutrient standards induce a lower level of enrichment than assumed (i.e., a reduced *k*, or growth coefficient), the time to nuisance biomass is extended and would lengthen the associated duration beyond 14 days. These two uncertainties will tend to counter-balance one another, and we concluded that 14 days is a good approximation of the central tendency of the modeled results.

We then considered the 14-day duration in the context of the results from a whole-stream enrichment study carried out by the Department in a Montana stream (Suplee and Sada de Suplee, 2011). In the enrichment study, peak algal biomass at the location in the study reach with the most algae—as documented by photo series and quantitative measurement—occurred about 20 days after N and P dosing began. Dosing was set at moderately-enriched levels and the algal biomass peaked at the location at 1,092 mg Chl*a*/m^2, a density nearly identical to the maximum value we assumed in **Equation 3**. (The biomass peak comprised filamentous algae, not diatoms.) Initial algal biomass in the stream at the study site was around 40 mg Chl*a*/m^2. The average stream water temperature over the time period was 21.8°C (range 16.2°C to 28.9°C), very close to the reference temperature of 20°C used in **Equation 3**. Note that the 20-day time-to-peak-biomass from the stream enrichment study closely aligns with the time-to-peak (T_{PB}) biomass in the modeled results (**Figure 2)**. Taken together, the laboratory studies (**Table 1)**, the modeled results **(Figure 2)**, and the results from the stream-enrichment study indicate that 14 days is an appropriate duration for nutrient control to maintain benthic algae below nuisance levels.

Recurrence frequency of low-flow events is the second consideration. The U.S. Environmental Protection Agency (USEPA) recommends a site-specific, biologically-driven approach for permitting discharges, where the average concentration of a toxic pollutant to which aquatic life can be chronically exposed without deleterious effects over a 4-day period should not occur more than once every 3 years (Stephan et al., 1985). Four days equates to the duration of exposure, once in three years the allowable recurrence frequency. In theory, this approach ensures excursions of toxic pollutants are uncommon enough that sufficient time passes for the aquatic community to recover in the interim years. Excess nutrient concentrations also lead to biological changes and impacts which then require time for recovery, therefore a recurrence frequency of about once every three years is probably a good starting point for nutrient pollution. Accordingly, we selected a seasonal (July 1to October 1) 14Q5 flow as the design flow for application to nutrient standards to be slightly protective. The 14-day duration reflects the time it can take to achieve nuisance biomass in wadeable streams and shallow parts of large rivers if nutrients are elevated (fewer days would be more-protective, more days less-protective). The 5-year recurrence frequency is close to USEPA's long-standing recommendation (i.e., once in three years) while being slightly protective. And, the seasonal 14Q5 flow is routinely reported by the U.S. Geological Survey (McCarthy, 2004), therefore it is readily available for use in MPDES discharge permits.

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To: Montana Board of Environmental Review

CC: Eric Urban, Water Quality Standards Section Supervisor

From: Michael Suplee, Ph.D., and Kyle Flynn, P.H., Environmental Science Specialists, Montana Dept. of Environmental Quality

Date: 3/19/2014

RE: Benthic algae biomass levels protective of fish and aquatic life in western Montana streams

In Suplee and Watson (2013), an algal biomass value of 125 mg chlorophyll *a*/m^2 is used in western Montana ecoregions to help derive the numeric nutrient criteria for those areas. Excessive bottom-attached algae in streams is a classic symptom of nutrient over-enrichment and controlling such algal growth is one of the objectives of numeric nutrient standards. This memorandum addresses how the 125 mg chlorophyll *a*/m^2 algae biomass value was determined by the Department.

Between 2009 and 2011, the Department carried out a whole-stream nitrogen and phosphorus addition study in a stream in southeastern Montana (Box Elder Creek). The project's objective was to determine what types of impacts to stream beneficial uses occur as a direct result of elevated nutrient concentrations. As documented in Suplee and Sada de Suplee (2011), bottom-attached (benthic) algae grew to much higher levels in the nutrient-dosed reaches than it did in the control reach, reaching a maximum of 127 mg chlorophyll *a*/m^2 of benthic algae[1] in the most highly dosed reach. One of the study's most notable findings was that this high biomass of benthic algae then senesced *en masse* at the end of the growing season, leading to declines in dissolved oxygen (DO) concentration to as low as 1.37 mg/L near the stream bottom. Low DO occurred in what we surmise to be a series of disconnected patches (pools or deposition zones) along the stream bottom; reaches with extremely low DO were found in areas where the stream had slower velocities. It was also documented that the only viable DO sink was the large volume of dead and decaying algae on the stream bottom (confirmed by visual observation and photographs), because water-column BOD_5 samples taken at the time were all less than detection[2]. Thus, elevated nutrient concentrations led to excessive benthic algal growth which, when it senesced and died at the end of the growing season, caused exceedences of the state's dissolved oxygen standards (DEQ, 2012).

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[1] This value represents the average for the reach. The reach average was determined from eleven replicate chlorophyll a measurements collected systematically at eleven transects spaced along the reach.

[2] Sediment oxygen demand (SOD) was not a significant DO sink in Box Elder Creek as evidenced by thenear-saturation levels of water-column DO observed in the control reach throughout the study.

Dissolved oxygen standards are intended to protect fish and associated aquatic life. The finding that unusually high levels of benthic algae can lead to seasonal crashes in DO in wadeable streams is important because it demonstrates a direct link between elevated nutrient concentrations, resultant algae growth, and probable harm to fish and aquatic life that is disconnected in time with the initial nutrient loadings. But because the study was undertaken in southeastern Montana, it was necessary to carry out additional analyses to see if its findings were applicable to western Montana streams, which are generally colder and have steeper gradients.

To explore the effect of water temperature on the DO impacts observed in the eastern Montana study, we developed a modified Streeter-Phelps (1925) analytical model of the Box Elder Creek nutrient dosing reach. The physical basis of the model was then used to evaluate the response of hypothetical streams in western Montana at an elevation of 1,219 m (4,000 ft), with mean daily water temperature of 7 ^{o}C (a common temperature in the region), and with varying physical characteristics (gradient and velocities, reflecting different reaeration behavior). Methods, discussion and recommendations are provided below.

1.0 Simulation of Nutrient Dosing Study Findings for Cooler Water Temperatures, Higher Dissolved Oxygen Saturation, and Differing Reaeration Rates

As documented in Suplee and Sada de Suplee (2011), benthic algae senesced *en masse* at the end of the growing season which led to observed DO levels as low as 1.37 mg/L near the stream bottom in the High Dose reach (HD reach) in a depositional area (glide) with steady laminar flow. It was also documented that the only viable DO sink was the large volume of dead and decaying algae on the stream bottom. Thus, senesced algae are a significant and important DO sink when it comes to eutrophication, and we refer to it here as “senesced algae oxygen demand” (SAOD). The terminology has been used to differentiate it from normal sediment oxygen demand (SOD) which is associated with the oxygen consuming properties of organic material in a stream’s bottom sediments.

The questions addressed in this section are:

1. *If the nutrient dosing study had been carried out in an identical western Montana stream but with cooler average temperatures and higher dissolved oxygen saturation, would impacts to Montana’s dissolved oxygen standards still have occurred?*

2. *In the hypothetical stream with cooler water temperatures and higher dissolved oxygen saturation simulated in No. 1 above, what would be the effect of differing reaeration rates on the dissolved oxygen concentrations?*

Continuously monitored data (by YSI 6600 sonde) at the headwaters of the Box Elder High Dose (HD) reach (triangle **A** in **Figure 1-1**, just upstream of the nutrient addition point) showed that DO was very close to saturation entering the HD reach (measured as 8.8 mg DO/L @ 12 cm off the bottom; water temp = 15.3^{o}C; elevation 921 m). By the time water flowed down to the HD reach YSI sonde, DO concentration 12 cm off the bottom had dropped to 1.37 mg/L. The residence time between the two points was about 20 min as identified through velocity measurements. Box Elder Creek is a Rosgen C4 channel (Rosgen, 1996).

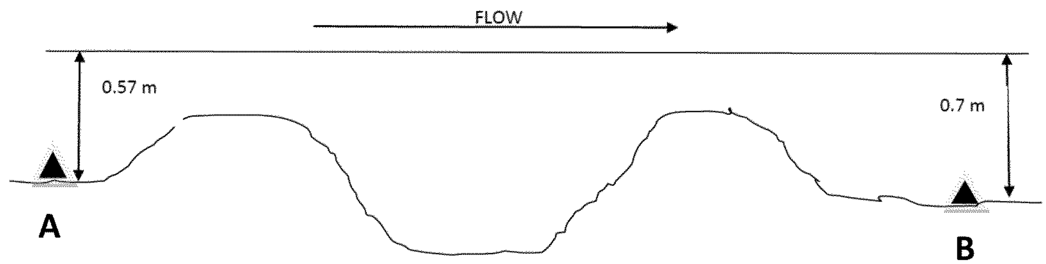

Figure 1-1. Longitudinal diagram of Box Elder Creek high-dose reach. The upstream YSI sonde (**A**, on left) was placed just upstream of the point where nutrients were added. There is a span of 110 m between it and the High Dose YSI sonde (**B**, on right). Two riffles and one pool were found in the reach between the two sondes.

Data indicate that it is unlikely that 1.37 mg DO/L persisted surface to bottom; rather, a vertical gradient in oxygen concentration probably occurred. Such a gradient is typical of steady-state conditions of diffusive mass transfer typified by Fick's first law. Given the mass of decaying algae observed, we inferred that DO was zero mg/L on the bottom of the channel whereas, at the surface, it was still probably at or near saturation (as observed upstream). A simple linear relationship was fit between the three DO-depth points (**Figure 1-2**; representing the linear change in oxygen over depth, i.e., dO_2/dz), and we believe a linear fit is very reasonable. Indeed, if poorly mixed (as typified in diffusion problems) a linear gradient would occur over the water column. In this instance, the linear model most reasonably fits the data.

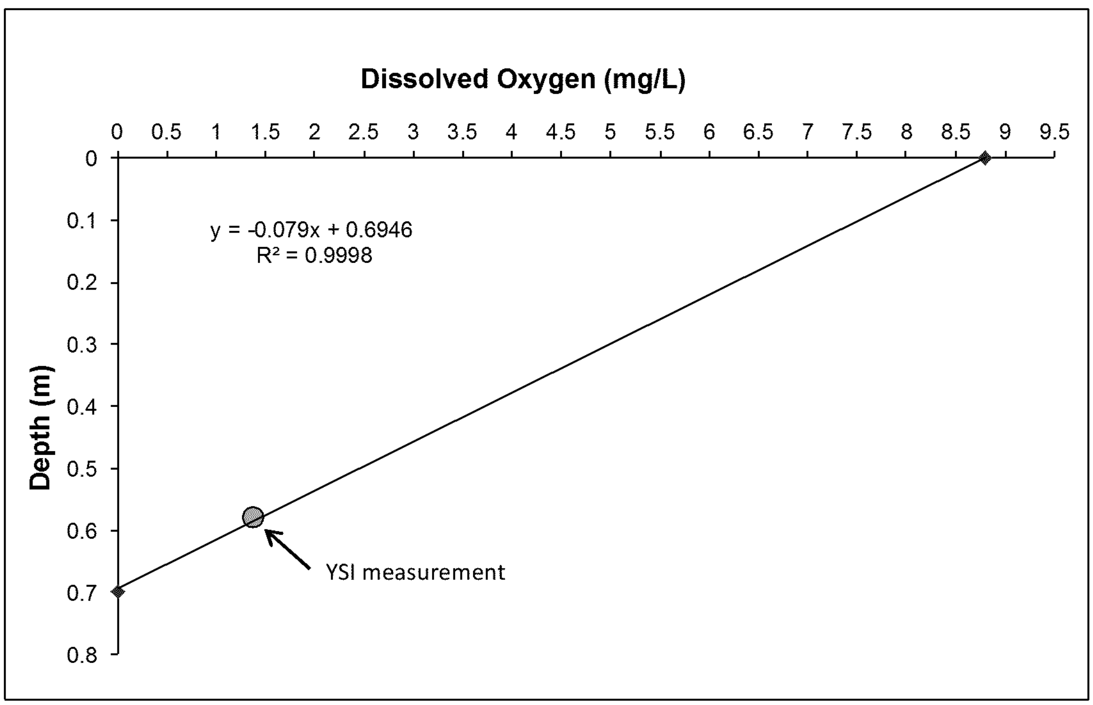

Figure 1-2. Estimated surface to bottom DO profile at the HD reach, 12:01 am, 10/6/2010.

Thus we have some knowledge about the oxygen gradient through the water column as well as the average oxygen concentration at the site in **Figure 1-2** given the previous assumptions. When using the linear assumption, the midpoint between 0 (bottom) and 8.8 (surface) mg DO/L is effectively the mean DO concentration of the site (i.e., 4.4 mg/L). An analytical solution to the Streeter-Phelps model with constant SAOD was then developed. With this new model we could evaluate whether differences in reaeration rates and algae growth rates between eastern and western Montana streams preclude (or do not preclude) the use of an algae level of 125 mg chlorophyll *a*/m^2 to assess aquatic life impacts in western Montana streams. The algae level was slightly lowered, from 127 to 125 mg chlorophyll *a*/m^2, to provide a threshold that is somewhat more protective; after all, we documented DO impacts at 127. Assuming plug flow (i.e., where advection is dominant) and a channel with uniform slope and cross-sectional area, the 1-D mass balance for DO over a differential element (Δ x) is

$$\Delta V \frac{DO}{dt} = J_{in} A_c - J_{out} A_c \pm \text{reaction}$$

where ΔV = volume of element (m^3), A_c= channel area (m^2), DO = dissolved oxygen concentration (g/m^3), and J_{in} and J_{out} = flux of DO in and out of element due to advection (g/m^2 d). Flux in and out of the element is defined as

$$J_{in} = U(DO)$$

$$J_{out} = U(DO + \frac{dDO}{dx} \Delta x)$$

where U =channel velocity (m/d) and Δx = incremental distance (m).

By adding first-order reaction rate (i.e., reaeration) and a zero-order term for senesced algae oxygen demand (SAOD [gO_2/m^2/d]), the equation looks like

$$\Delta V \frac{DO}{dt} = U(DO) A_c - U(DO + \frac{\partial DO}{\partial x} \Delta x) A_c + k_a \overline{(DO_{sat} - DO)} \Delta V - \frac{SAOD}{H} \Delta V$$

where k_a = first-order rearation coefficient (/d), DO_{sat}-DO = the average DO deficit, and H = channel depth (m).

Collecting terms, dividing by $\Delta V = A_c \Delta x$, and taking the limit as $\Delta x \rightarrow 0$ yields

$$\frac{DO}{dt} = -U \frac{\partial DO}{\partial x} + k_a \overline{(DO_{sat} - DO)} - \frac{SAOD}{H}$$

At this point, it should be noted that the first-order reaeration rate k_a is actually a function of the liquid mass transfer velocity [k_l, m/d] divided by H, which relates atmospheric flux to surface area. Also, because of the high Henry's constant of O_2, and the fact that oxygen in the atmosphere is constant, exchange is strongly liquid-film controlled. Consequently, DO at saturation (DO_{sat}) can be well characterized by altitude and temperature only. Finally, to simplify the differential equation, we reformulate DO concentration as dissolved oxygen deficit $D = DO_{sat}$-DO [mg O_2/L], which switches the sign of the DO input/output terms. Under steady state conditions the temporal derivative goes away and we are left with

$$0 = -U\frac{dD}{dx} - k_a D + \frac{SAOD}{H}$$

The above differential equation can then be solved by using integrating factors and yields the equation below, where D_0=the initial DO deficit (mgO_2/L) and x equals the distance downstream (m) from initial conditions:

$$D = e^{-\frac{k_a}{U}x} \left(D_0 + \frac{SAOD}{Hk_a} \left(e^{\frac{k_a}{U}x} - 1\right)\right)$$

Finally, substitution of k_a with the approximation from Owens et al. (1964) makes the equation appropriate for small streams (Covar, 1976) (in metric units, where U is in m/s):

$$k_a = 5.32\frac{U^{0.67}}{H^{1.85}}$$

Thus the equation is applicable to small shallow streams where the oxygen generation and consumption processes are primarily reaeration and SAOD. It should be noted that reaeration is temperature adjusted using the Arrhenius equation with a theta (θ) of 1.024 (Chapra, 1997). Also, since we have omitted respiration and photosynthesis from our equation, the results are probably only appropriate for night-time conditions only. Finally, note that algal respiration does consume oxygen in dark reactions through carbon oxidation, but it was not included in the model.

Following model development, we then calibrated the analytical model to data from Box Elder Creek to estimate SAOD expected from an accumulation of dead/decaying algae, which could then be transferred to other streams. In this instance the calibrated SAOD was very high, approximately 92 gO_2/m^2/day. Using the Arrhenius equation and a theta (θ) of 1.047 (applicable to BOD decomposition; Chapra, 1997), this zero-order rate equates to 76 gO_2/m^2/day at a stream temperature of 7°C. In other words, the biological decomposition rate of the dead algae determined from the Box Elder Creek dosing study has been reduced mathematically to a colder water temperature. We refer to this new, adjusted SAOD as $SAOD_{COLD}$.

Figure 1-3 below compares the model output for Box Elder Creek vs. the simulated, carbon-copy western Montana stream evaluated through the model. Conditions were kept the same except that the western-Montana simulation was colder (7° C, vs. 15.3° C in Box Elder Cr.) and at a higher elevation (1,219 m vs. 921 m at Box Elder Cr.).[3] In general, we see a longitudinal decline in DO concentration

[3] Velocity in both streams was set at 0.09 m/s, depth at 0.27 m, as measured in Box elder Cr. We also used measured DO (8.8 mg/L) for initial conditions in Box Elder Cr. as observed just upstream of the nutrient-dosed reach (at the sonde shown as A in Figure 1-1); DO at saturation in Box Elder Cr., at 15.3°C and the site elevation, is very close (8.97 mg/L). In the western-MT simulation, DO was set at 10.4 mg/L (saturation at the temp. and elevation given). The ka values for Box Elder Creek and its western-MT simulation were 11.1/day and 9.1/day, respectively.

reflective of a waterbody flowing over a very large diffuse SAOD source. The zero-order SAOD used for the western MT stream is $SAOD_{COLD}$. The model output presents average water column DO (i.e., the midpoint of the surface-to-bottom DO gradient in **Figure 1-2**) longitudinally, and shows how that average would decline over space due to SAOD. Note that the DO standard is actually exceeded sooner in the western Montana simulation (occurring 75 m downstream as opposed to 100 m in Box Elder Creek). This happens because the western Montana DO standard is set at a higher concentration (8 mg DO/L vs. 5 mg DO/L in much of eastern Montana) because it is intended to protect salmonid fishes.

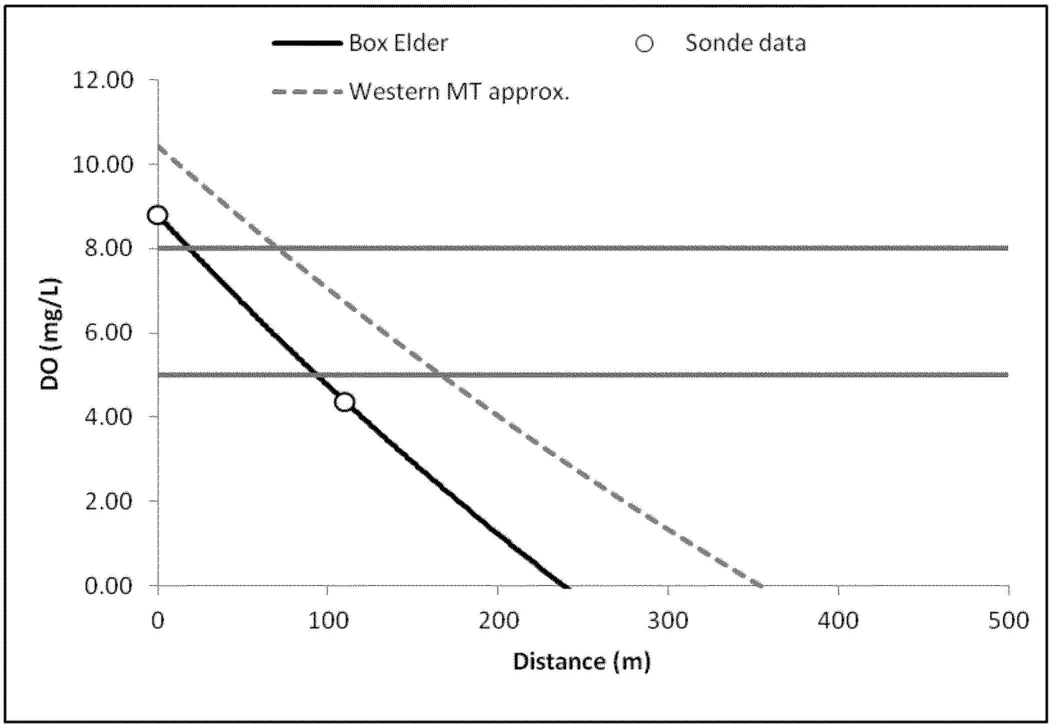

Figure 1-3. Model output for two scenarios: (1) Box Elder Creek nutrient dosing study, including YSI sonde data from the HD reach which were used to calibrate the model, and (2) the simulation of an identical stream but at 7^{o} C and at 1,219 m elevation. Red horizontal lines show the DO standards typical for each region (lower red line for eastern MT, upper red line for western MT).

1.1 Simulations Using Different Reaeration Rates

To evaluate the potential effect of the calibrated SAOD presented above, we evaluated a number of stream types commonly encountered in western Montana. The primary difference between the evaluated streams and the Box Elder Creek calibration was the dependence of reaeration rate on channel configuration, and (as before) the effect of colder temperatures and higher altitude on oxygen saturation and biological decomposition rates. Western Montana streams were segregated using the Rosgen stream classification system which integrates factors such as slope, channel width/depth ratio, substrate, etc. Given the dependence of the reaeration coefficient on channel depth and velocity, Manning's equation was used to determine effective velocities for various channel configurations:

$$U = \frac{1}{n} R^{2/3} S^{1/2}$$

where U = velocity in m/sec, R = hydraulic radius in m, S = water surface slope in m/m (Dunne and Leopold, 1978, but in SI units), and n = Manning's coefficient. Rosgen (1996) provides descriptive statistics for many of his stream types (e.g., C4, F4 channels), and values (e.g., surface water slope, cross-sectional area) representing the central tendency of each group were selected and input into Manning's relationship in order to derive a representative velocity for the stream class group (**Table 1-1**). Roughness coefficients between 0.05 and 0.06 were selected which, from our experience, are reflective of streams during low-flow conditions (recall that the roughness coefficient actually is not independent of flow and depth). Chow (1959) reports variation in Manning's n with stage and also that weeds (i.e., macrophytes and attached algae) in stream channels induce somewhat higher Manning's n values. Given that our simulated streams would have fairly thick mats of filamentous algae, slightly higher-than-textbook Manning's n values are well justified[4].

Table 1-1. Stream Channel Characteristics for Different Representative Rosgen Stream Channels Used in the Model.

Rosgen Stream Type	X-sectional area (m^3)	Width (m)	Depth (m)	Surface Water Slope (m/m)	Manning's n used	Velocity (m/sec)
B4	1.54	5.77	0.27	0.0200	0.05	1.11
C3	3.09	11.55	0.27	0.0023	0.06	0.32
C4	2.70	10.09	0.27	0.0045	0.06	0.45
C5	2.51	9.39	0.27	0.0005	0.06	0.14
E3	0.88	1.16	0.76	0.0100	0.06	0.80
E4	0.54	0.71	0.76	0.0100	0.06	0.65
E5	0.56	0.73	0.76	0.0010	0.06	0.21
F4	1.95	7.31	0.27	0.0018	0.06	0.28
G4	0.74	2.78	0.27	0.0200	0.06	0.87

The calculated velocities and depths for the stream groups (**Table 1-1**) were used to recalculate the reaeration coefficient in the model and evaluate the effects on longitudinal DO decline. As before, the simulations were run under the assumption that the stream, irrespective of whether it was C4, B4, etc., was at 7^{o} C and at an elevation of 1,219 m. We tested Rosgen B, C, E, F, and G stream types. Following initial testing, it was clear from the B-channel results that the higher gradient A-channels did not need to be evaluated since their gradients/velocities would overcome SAOD. We did not test D channels as Rosgen (1996) provides no descriptive statistics. It was found that C and F channels would be vulnerable to low oxygen problems (i.e., their velocities were insufficient to overcome $SAOD_{COLD}$), the Rosgen C4 channel somewhat less so than C3 or C5. In contrast, A, B, E, and G channels would probably not be impacted by low DO. It is not clear whether D channels would be vulnerable to DO problems or not; some of the lower gradient ones very likely would. **Figure 1-4** below contrasts results from three examples; a hypothetical B4 channel, C3 channel, and F4 channel. As can be seen, high velocities in the B4 channel lead to higher reaeration rates capable of overcoming the $SAOD_{COLD}$; in contrast, the C3 and

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[4] Box Elder Creek, which is a Rosgen C4 channel with a D_{50} of 45 mm (coarse gravel), required an even higher Manning's n (0.08) in order to match the Manning's equation result tomeasured average stream velocity and slopes. Thus, the use of 0.06 in our calculations for Rosgen stream types is quite reasonable.

F4 channel become impacted (relative to their DO standards) in about the same way Box Elder Creek was.

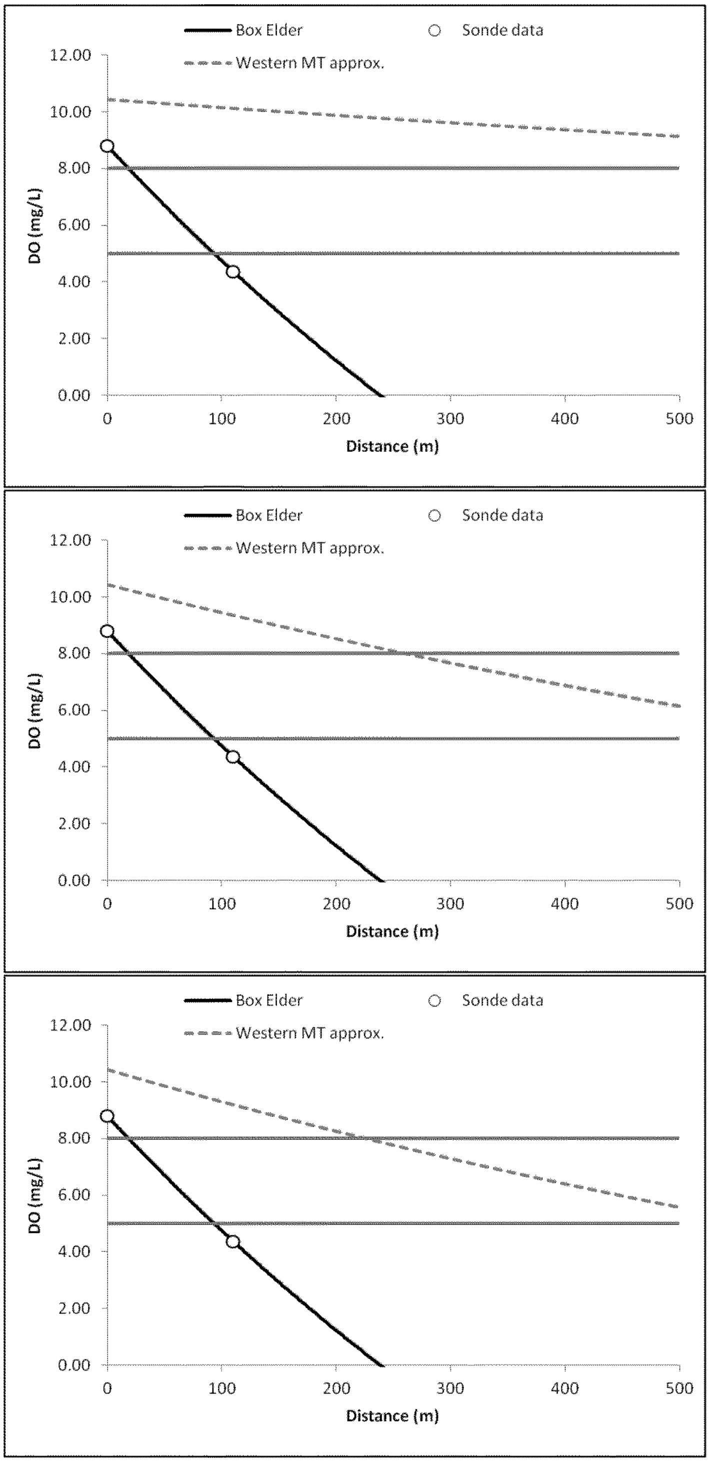

Figure 1-4. Modeled DO impacts as a function of Rosgen stream class. Upper Panel. Rosgen B4 channel. Dissolved oxygen would not fall below the 8.0 mg DO/L standard for a long distance and DO impacts are almost certainly negated. Middle Panel. Rosgen C3 channel. Violations of the DO standard occur within 300 m. Lower panel. Rosgen F4 channel. Dissolved oxygen impacts are similar to the C3 channel, but occur in an even shorter distance.

1.2 Discussion

Others discuss end-of-growing season senescence of plants and its effect on stream water quality. Jewell (1971) notes in streams in England that "At the end of the growing season, or when the weeds are killed, their decomposition may exert heavy demands on the oxygen resources of a water". Novotny and Bendoricchio (1989) observe that "oxygen deficiency is highest and most troublesome in streams where shallow productive zones are followed be deeper sections". The latter statement largely conforms to what we observed, where senesced algae accumulated in the glides and pools of the nutrient-dosed Box Elder Creek reach, and these areas manifested discontinuous areas of low DO longitudinally along the reach.

The reach-average level of benthic algae leading to the DO problem at the Box Elder Creek HD reach (127 mg Chl*a*/m^2) has been observed in eutrophied streams in western Montana, and quite often *Cladophora* provides a substantial proportion of this biomass (just as was observed in the Box Elder study). Thus, a key biological characteristic of the plains stream we studied, in terms of the biomass and algae type, is quite comparable to western Montana streams. Therefore, we believe it can be reasonably concluded that if the study had been carried out in an identical stream in western Montana, but at 7^{o} C and with DO at saturation of 10.4 mg/L, one would have seen exceedences of the DO standards following a similar pattern.

The SAOD we calculated is far higher than sediment oxygen demand (SOD) reported in the literature (highest SOD located was 21.4 g O_2/m^2/day; Ling et al., 2009). But as mentioned at the start, SAOD is not SOD in the normal sense and the rates are not strictly comparable. Novotny and Olem (1994) report that feedlot runoff (a highly organic, putrescible material) can have BOD_5 of 1,000-12,000 mg/L. If this material were all to settle to the stream bottom in a stream having the same average depth as Box Elder Creek and then exert its DO demand, BOD_5 values of this magnitude would equate to 53-640 g O_2/m^2/day (at 20^{o} C). The SAOD we calculated for decomposing benthic algae (equal to 133 g O_2/m^2/day @ 20^{o} C) clearly falls to the low side of this range and is, therefore, a reasonable estimate. It should also be noted that the true SAOD may be higher, but much more localized spatially along the reach (i.e., we calibrated it assuming SAOD over the entirety of the reach).

In conclusion, it appears that the Box Elder dosing study, had it been carried out under identical circumstances except for lower water temperature and higher DO saturation, would have led us to similar conclusions about the impacts of senesced algae oxygen demand and associated effects on stream DO dynamics (**Figures 1-2, 1-3**). However, our simulations indicate that the gradients of some western Montana streams are sufficiently high that SAOD can, for all practical purposes, be overcome (**Figure 1-4**, upper panel). Based on our findings, we recommended the following:

1. The 125 mg Chla/m^2 threshold should apply to all Rosgen C and F channels, as their group characteristics (velocity, depth, etc.) appear to be insufficient to overcome DO impacts from senesced algae oxygen demand;
2. The 125 mg Chla/m^2 threshold would not apply to Rosgen A, B, E and G channels, as their group characteristics (velocity, depth, etc.) appear to be sufficient to overcome DO impacts from senesced algae oxygen demand; and

3. No recommendation is made for Rosgen D channels at this time. These channels are not so commonly encountered in Montana and cases will need to be evaluated case-by-case.

Note that in streams where the 125 mg Chl*a*/m^2 threshold does not apply, the recreationally-derived benthic algal threshold (150 mg Chl*a*/m^2) does. Because 125 mg Chl*a*/m^2 links directly to DO impacts, it is associated with the fish and associated aquatic life beneficial use; in contrast, 150 mg Chl*a*/m^2 applies to the recreation use (per Suplee et al., 2009).

2.0 Final Department-recommended Benthic Algae Threshold

A meeting was held between management and technical staff in March 2012 to consider these findings and their implication for making stream assessment decisions. Careful consideration was given to the totality of information provided by the Box Elder nutrient dosing study, the modeling results discussed above, and the benthic algae threshold for protecting the recreation use (150 mg Chl*a*/m^2). Our analysis showed that some western Montana streams would be vulnerable to low DO problems if benthic algae reached 127 mg Chl*a*/m^2, but other streams would not. There was relatively little difference in magnitude between the level of algae that would prevent impacts to fish and aquatic life (~125 mg Chl*a*/m^2) vs. the level which protects recreational (150 mg Chl*a*/m^2). Further, monitoring staff who have completed many stream assessments involving benthic algae have indicated that in the vast majority of cases benthic algae levels are either well below or well above the thresholds in question (i.e., borderline cases near the thresholds that would a require more detailed analysis and data collection are uncommon). Thus, in order to preclude a complex, two-threshold system requiring Rosgen stream class identification in all cases, the Department instead decided that in western Montana streams a single benthic Chl*a* threshold of 125 mg Chl*a*/m^2 (site average) would be used. Values above this threshold are considered impacts to both the aquatic life and the recreational beneficial uses, as documented in Suplee and Sada de Suplee (2011). And as noted at the start of this memorandum, 125 mg Chl*a*/m^2 has since been used to help derive the proposed base numeric nutrient standards for western Montana ecoregions (Suplee and Watson, 2013; draft Department Circular DEQ-12A).

3.0 References

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